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CARBON FOOTPRINTING

Assessment of carbon sustainability under different tillage systems in a double rice cropping system in Southern China

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Abstract

Purpose Adoption of the carbon (C)-friendly and cleaner technology is an effective solution to offset some of the anthropogenic emissions. Conservation tillage is widely considered as an important sustainable technology and for the development of conservation agriculture (CA). Thus, the objective of this study was to assess the C sustainability of different tillage systems in a double rice (*Oryza sativa* L.) cropping system in southern China.

Methods The experiment was established with no-till (NT), rotary tillage (RT), and conventional tillage (CT) treatments since 2005. Emission of greenhouse gasses (GHG), C footprint (CF), and ecosystem service through C sequestration in different tillage systems were compared.

Result and discussion Emission of GHG from agricultural inputs (Mg CO_2-eq ha^{-1} year^{-1}) ranged from 1.81 to 1.97 for the early rice, 1.82 to 1.98 for the late rice, and 3.63 to 3.95 for the whole growing season, respectively. The CF (kg CO_2-eq kg^{-1} of rice year^{-1}) in the whole growing seasons were 1.27, 1.85, and 1.40 [excluding soil organic carbon (SOC) storage] and 0.54, 1.20, and 0.72 (including SOC storage) for NT, RT, and CT, respectively. The value of ecosystem services on C sequestration for the whole growing seasons ranged from ¥3,353 to 4,948 ha^{-1} year^{-1} and followed the order of NT > CT > RT. The C sustainability under NT was better than that under RT for the late, but reversed for the early rice. However, NT system had better C sustainability for the whole cropping system compared with CT.

Conclusions Therefore, NT is a preferred technology to reduce GHG emissions, increase ecosystem service functions of C sequestration, and improve C sustainability in a double rice cropping region of Southern China.

Keywords Carbon footprint · Carbon sustainability · Conservation tillage · Ecosystem services · Paddy soils · Southern China

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1 Introduction

Climate change, an increasingly important focus of scientists and politicians, has become an important issue and a challenge for the humankind. Agriculture is one of the principal sources of greenhouse gas (GHG) emissions globally (Intergovernmental Panel on Climate Change 2013). However, agriculture can also be an important solution to mitigate climate change by reducing the net GHG emissions from the manufacture and use of agricultural inputs and by sequestering atmospheric CO_2 in soil and biota (Lal 2004b). Therefore, coordinated and focused efforts are needed to explore and develop appropriate technologies related to cleaner agricultural production and sustainable development for mitigating GHG emissions (Lal 2004b; Zhang et al. 2013b). Adoption of recommended management practices (RMPs) (e.g., conservation tillage) can reduce the carbon (C) footprint (CF) of agriculture, offset anthropogenic emissions (Stavi and Lal 2013), and enhance the ecosystem services (Lal et al. 1999; Lal 2013).

The magnitude of reduction of the CF of agriculture depends on the degree of reduction of GHG emissions from all agricultural inputs. The CF can be assessed by using a life cycle approach (LCA), which is an environmental impact assessment methods and has been widely applied to agricultural production systems (Andersson et al. 1994; Haas et al. 2000; Brentrup et al. 2004). The CF of agricultural produce is used to identify low C options and technologies and assess their mitigation potential (Gan et al. 2012a). Optimization of farming practices (e.g., improving N management, adopting RMPs) can effectively reduce the CF of agricultural products (Gan et al. 2012a, b, c). Taking sequestration of soil organic carbon (SOC) into account, the CF of spring wheat (*Triticum aestivum* L.) under a range of cropping systems has been determined in Canadian prairies to assess the effect of agricultural activities and related farming practices on environmental quality (Gan et al. 2012a).

The GHG emissions can be used to evaluate CF as an environmental performance indicator (Weinheimer et al. 2010) and also an economic indicator affecting ecosystem services (Costanza et al. 1997). Agricultural practices have large environmental impacts and also affect a series of ecosystem services (e.g., C sequestration) (Dale and Polasky 2007). Previous studies on agro-ecosystem services have mainly been conducted at macro level with relatively coarse spatial resolutions (Costanza et al. 1997), and only a few studies have been performed on a field experimental scale (Xiao et al. 2005; Schipanski et al. 2014). A widespread adoption of RMPs is essential to sustaining or improving ecosystem services and reducing tradeoffs (Lal 2013). Ecosystem services are the benefits people obtain from ecosystems (Costanza et al. 1997), and those associated with the C cycle can contribute to mitigating climate change by enhancing C storage and sequestration in the terrestrial biosphere (Raupach 2013). Therefore, C sequestration is a strong determinant of ecosystem services. Optimization of SOC concentration is conducive to improving ecosystem functions, such as increasing crop yield, offsetting anthropogenic emissions, and strengthening the coupled cycling of numerous elements (Lal 2013). The conservation agriculture (CA) system is meritorious in the provisioning of diverse ecosystem services, such as climate mitigation through SOC sequestration and reducing GHG emissions (Palm et al. 2014). Sustainability of a production system can be assessed through evaluating temporal changes in the net C output to C input ratio (Lal 2004b). It is, thus, necessary to objectively assess the value of specific ecosystem services (e.g., C sequestration) at a field scale (Lal 2013).

Rice (*Oryza sativa* L.) is one of the most important grain crops in China, and there exists a large potential of C sequestration in soils cultivated to rice (Pan et al. 2004). However, paddy rice is also an important source of GHG emissions (e.g., CH_4 and N_2O) (Hou et al. 2000). Several studies have focused on the assessment of GHG emissions from paddy fields in the context of mitigating climate change. Only a few studies have assessed GHG emissions in rice production using LCA (Hou et al. 2000; Kruger and Frenzel 2003; Blengini and Busto 2009; Thanawong et al. 2014). Tillage and crop residue retention strongly impact GHG emissions from farmland due to the change in soil properties and numerous biochemical processes within the soil solution (Al-Kaisi and Yin 2005). The double paddy system is critical to the national food security and accounts for ~40 % of the total rice production in China (Bai 2013). In addition, inappropriate farm managements (e.g., burning crop residues and long-term flooding) exacerbate GHG emissions from paddy soil in southern China. Therefore, it is necessary to assess the C sustainability of all farm operations to identify a C-friendly technology for this region. Conservation tillage is a RMP which can moderate GHG emissions, increase SOC sequestration, and improve ecosystem services of a cropping system (Lal 2013; Zhang et al. 2013a, b). To date, a few studies have been conducted to assess C sustainability by assessing the GHG emissions, CF, and the value of C service function synthesizing economic and ecological efficiency in double rice cropping systems for a range of tillage practices. Thus, it is important to explore and develop C-friendly and sustainable tillage technologies for reducing agricultural GHG emissions and increasing SOC sequestration in paddy-based cropping systems in China. Therefore, the objective of this study was to assess the agricultural GHG emissions, CF, and C sustainability for different tillage practices and identify optimal and C-friendly tillage systems for the double rice cropping regions in Southern China.

2 Methods

The data presented in this article were based on field experiments. The relevant data of agricultural inputs for the period from 2006 to 2011 used in this article were collated and synthesized from the published literature.

2.1 Experimental site description

The experiment was initiated in 2005 in the Ningxiang County (28° 07′ N, 112° 18′ E, elevation 36.1 m) of Hunan province, Southern China. The site has a subtropical monsoonal humid climate with a mean annual temperature of 16.8 °C, an average annual rainfall of 1,360 mm, and an annual sunshine duration of 1,740 h. Predominant soil of the experimental site is classified as Stagnic Anthrosols developed from the Quaternary red earth (Gong et al. 2007). Principal physical and chemical properties of 0–20-cm depth are as follows: 34.9 g kg^{-1} of soil organic matter (SOM), 1.21 Mg m^{-3} of bulk density, 1.29 g kg^{-1} of total nitrogen, 1.23 g kg^{-1} of

total phosphorus, 17.63 g kg^{-1} of total potassium, 224 mg kg^{-1} of available nitrogen, 4.0 mg kg^{-1} of available phosphorus, 97.1 mg kg^{-1} of available potassium, and pH of 6.3. Double cropping, comprising of the early and late rice, is the predominant cropping system in this region. Prior to transplanting of both early- and late-season rice, rotary tillage was the principal tillage method since 1990s, and plow tillage was widely used before then. Rice is transplanted by throwing the seedling in the standing water since 1996. The rice residues are commonly burnt after harvest in both early and late seasons.

2.2 Experimental design and management

The 6-year experiment was laid out according to a randomized complete block design with three replications, with the plot size of 66.7 m^2 (8.34×8 m). Three tillage treatments included the following: no-till (NT), conventional tillage (CT), and rotary tillage (RT). For each treatment, rice residues were retained on the soil surface after harvest in both seasons. No additional tillage was performed on the NT plots, and rice residues were retained as surface mulch throughout the cropping period. For CT and RT plots, water was ponded to ~2-cm depth to facilitate tillage operations, which were performed prior to transplanting of rice seedlings in both seasons. Rice residues were incorporated into the soil in these plots. The CT plots were tilled once with a moldboard plow to a depth of 15 cm and then rotovated twice to a depth of 8 cm before transplanting of rice seedlings. The RT plots were rotovated four times to a depth of 8 cm before transplanting of rice seedlings. Diesel use for both early and late rice was ~27 and 21 L ha^{-1} to plow and rotate the field, respectively (the density of diesel is 0.85 kg L^{-1}).

Details of agricultural inputs in the experiment from 2006 to 2011 were depicted in Table 1. Rice seeds were soaked in water for 12 h to facilitate germination. Nursery beds of rice seedlings were leveled with a plank, then trays (0.05 kg, 308 holes, of 60×33×2.5-cm dimension) filled with soil slurry were put on the seedling bed. Rice seeds were broadcasted uniformly on the surface of trays (1,200 individual ha^{-1}) at the seeding rate of 120 and 90 kg ha^{-1} for early and late rice. Nursery was seeded during the middle 10 days of March and June, respectively. Trays were made of polyvinyl chloride and were used twice for raising seedling of the early and late rice. The early and late rice were transplanted by manually throwing of rice seedlings in April and July and combine harvested in July and October, respectively. The diesel consumption for combine harvesting was ~35.7 L ha^{-1} for all treatments. The cultivars for early rice were Yizao9 in 2006 and Zhongjiazao32 from 2007 to 2011. Similarly, cultivars for late rice were Xiangwanshan13 from 2006 to 2009 and Zhuliangyou2 in 2010 and 2011. Density of transplanting was ~3.6×10^5 and 2.7×10^5 hills ha^{-1} for the early and late rice each year, respectively. All plots received 688.5 kg ha^{-1} of compound fertilizer ($N:P_2O_5:K_2O=18:5:6$) and 78 kg ha^{-1} of urea (46 % N) for the early rice, 694.5 kg ha^{-1} of compound fertilizer ($N:P_2O_5:K_2O=18:5:6$) and 87 kg ha^{-1} of urea for the late rice as basal fertilizer at transplanting in 2006. From 2007 to 2011, all plots received 375 kg ha^{-1} of compound fertilizer ($N:P_2O_5:K_2O=$ 20:12:14) as basal fertilizer at transplanting, followed by 150 kg ha^{-1} of urea ~1 week later for the early rice, 375 kg ha^{-1} of compound fertilizer ($N:P_2O_5:K_2O=$ 20:12:14) and 75 kg ha^{-1} of urea as basal fertilizer at seedling transplanting, followed by 75 kg ha^{-1} of urea ~1 week later for the late rice. Thus, total amounts of N, P_2O_5, and K_2O for early and late rice were ~155, 34, and 41 kg ha^{-1} in 2006 and ~144, 45, and

Table 1 The amount of agricultural inputs under different tillage systems in the double rice cropping system (kg ha^{-1} $year^{-1}$)

Component	Early rice			Late rice			The whole growing season		
	NT[a]	RT	CT	NT	RT	CT	NT	RT	CT
Tillage[b]	0[c]	17.9	31.9	0	17.9	31.9	0	35.7	63.8
Harvest[b]	35.7	35.7	35.7	35.7	35.7	35.7	71.4	71.4	71.4
Irrigation[b]	12.8	12.8	12.8	22.1	22.1	22.1	34.9	34.9	34.9
Tray	30.0	30.0	30.0	30.0	30.0	30.0	60.0	60.0	60.0
Fungicide	0.2	0.2	0.2	0.7	0.7	0.7	0.9	0.9	0.9
Herbicide	1.2	1.2	1.2	1.2	1.2	1.2	2.4	2.4	2.4
Insecticide	1.5	1.5	1.5	2.3	2.3	2.3	3.8	3.8	3.8
Urea	138.0	138.0	138.0	138.0	138.0	138 .0	276.0	276.0	276.0
Compound fertilizer	436.2	436.2	436.2	436.2	436.2	436.2	872.4	872.4	872.4
Seed	120.0	120.0	120.0	90.0	90.0	90.0	210.0	210.0	210.0

[a] *NT* no-till, *CT* conventional tillage, and *RT* rotary tillage

[b] The diesel consumption for tillage, harvest, and irrigation are calculated by multiplying the amount of actual application by the diesel density

[c] The quantity of agricultural inputs application are the mean value from 2006 to 2011

52.5 kg ha^{-1} from 2007 to 2011, respectively. Prior to transplanting of both early and late rice, paraquat (1,1′-dimethyl-4,4′bipyridinium ion) was applied to control weeds in all treatments at the rate of 1.2 kg ha^{-1} for both the early and late rice. Insecticides and fungicides were used to prevent disease and insect infestation as per the recommended practice in the rice growing period. Quantity of insecticides and fungicides used in the calculation of LCA were 1.5 and 0.2 kg ha^{-1} for early and 2.3 and 0.6 kg ha^{-1} for late rice, respectively. Paddy fields were flooded to ~40–50-mm depth for the entire growing seasons in all treatments, except during the aeration stage and 1 week before harvest, but were not irrigated during the winter fallow. In addition to rainfall, paddy was irrigated ~4 times (~3,000 m^3 ha^{-1}) for the early rice and ~8 times (~5,400 m^3 ha^{-1}) for the late rice to meet the evaporative demand. The water was pumped to irrigate paddy field by diesel-powered pump from a nearby river, and ~15 and 26 L ha^{-1} of diesel were consumed for the early and late rice, respectively.

2.3 Data collection and analysis

Estimates of GHG emissions from agricultural inputs and paddy fields were assessed for the entire production chain for both early and late rice. The system boundary of this study included the entire life cycle from cradle (agricultural inputs production from raw materials) to farm gates (harvested rice) (Fig. 1). The GHG emissions in double rice production system comprised of the following: (1) production, storage, and transportation of agricultural inputs (e.g., fertilizers, pesticides, seeds, etc.) to the farm gate and their applications; (2) total seasonal emissions of CH_4 and N_2O in paddy fields; and (3) energy consumption of machine operations including tillage, irrigation, and harvesting. Energy from human labor was not computed, because human respire CO_2 regardless of whether they are working or not (West and Marland 2002).

2.3.1 The GHG emissions from agricultural inputs

GHG emissions from agricultural inputs were calculated by Eq. (1):

$$E_{\text{input}} = \sum_i Q_{\text{emission}_i} \times \varepsilon_i \tag{1}$$

Where E_{input} is the total amount of GHG emissions from all agricultural inputs (kg CO_2-eq ha^{-1} year^{-1}); Q_{emission_i} is the amount of a *i*th individual agricultural input applied in cropping system (kg ha^{-1} year^{-1}), including fertilizer, diesel, tray, pesticides, and seed; ε_i is the specific emission rate of individual agricultural input to GHG emission including manufactured and/or application (kg CO_2-eq kg^{-1}). The background data in the research was used from the IKE eBalance v3.0 (IKE Environmental Technology CO., Ltd, China) based on the site-specific characteristics in China, the inventory database of which included the Chinese Life Cycle Database (CLCD, Sichuan university, China), European reference Life Cycle database (ELCD, Joint Research Centre, European Commission), and Ecoinvent database (Swiss Centre for Life Cycle Inventories, Switzerland). The specific emission rate of CO_2 equivalent for most of inputs were used from CLCD, and of pesticides and seeds were used from Ecoinvent (Table 2). Here, the specific emission rate for all agricultural material inputs indicated the amount of GHG released during the life cycle per unit product. According to above datasets, the quantity of actual consumptions for these agricultural inputs were used to calculate the GHG emissions from this experiment.

2.3.2 Calculation of carbon footprint

The CF of rice is mainly determined by the grain yield and total GHG emissions associated with the double cropping. The latter include emissions from agricultural inputs and non-CO_2 GHG emissions (e.g., N_2O and CH_4) arising from paddy fields.

Fig. 1 System boundary for calculating GHG emissions in the double rice cropping system

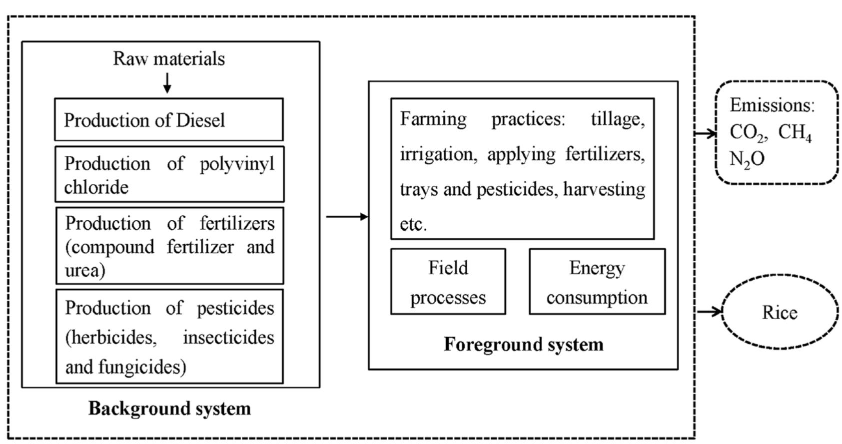

Table 2 The specific emission rate for different agricultural material inputs

Input	CO_2 equivalent ($kg\ CO_2\text{-eq}\ kg^{-1}$)	Sources[a]
Compound fertilizer	1.77	CLCD 0.7
Urea	2.39	CLCD 0.7
Diesel	0.89	CLCD 0.7
Diesel combustion	4.10	CLCD 0.7
Polyvinyl chloride	6.91	CLCD 0.7
Insecticides	16.61	Ecoinvent 2.2
Herbicides	10.15	Ecoinvent 2.2
Fungicides	10.57	Ecoinvent 2.2
Rice seed	1.84	Ecoinvent 2.2

[a] Parameters from CLCD (Chinese Life Cycle Database, 2012) and Ecoinvent 2.2 (2010)

The SOC storage is an important factor influencing CF of cropping systems, it changes significantly over time and varies substantially among cropping systems (Gan et al. 2012a). The annual soil C gain or loss for each tillage system was computed by the following Eq. (2) (Gan et al. 2012a):

$$\Delta SOC = \frac{SOC_{2011} - SOC_{2005}}{6year} \times \frac{44}{12} \tag{2}$$

Where, ΔSOC is the annual change in SOC storage in 0–30-cm depth since 2005 ($kg\ CO_2\text{-eq}\ ha^{-1}\ year^{-1}$), the SOC storage for different tillage practices were simulated with DNDC model (Huang et al. 2012). Terms SOC_{2011} and SOC_{2005} refer to the storage of SOC in the 0–30-cm layer after the late rice was harvested in 2011 and 2005, respectively; six is the duration of the study period in year, and 44/12 is the coefficient for converting C into CO_2.

Values of CFs for each tillage treatment were calculated using Eq. (3) for the early and late rice, the entire rice growing period, and with and without consideration of the changes in SOC stock during the study duration (Gan et al. 2012a).

$$CF = \frac{E_{input} + E_{CH_4} + E_{N_2O} + \Delta SOC}{Y} \tag{3}$$

Where CF is a carbon footprint of a treatment ($kg\ CO_2\text{-eq}\ kg^{-1}\ year^{-1}$ of grain), Y is the grain yield of rice ($kg\ ha^{-1}\ year^{-1}$), and ΔSOC is the change in the amount of SOC storage ($kg\ CO_2\text{-eq}\ ha^{-1}\ year^{-1}$) when this factor was included in the CF calculation. Terms E_{N_2O} and E_{CH_4} are the amount of cumulative non-CO_2 GHG emissions (CH_4 and N_2O) converted to CO_2 equivalent ($kg\ CO_2\text{-eq}\ ha^{-1}\ year^{-1}$) from paddy fields in the rice growing seasons, respectively, which were compiled from the available literature (Wu et al. 2008; Zhang et al. 2013a; Cui et al. 2014), which were measured by using the closed static chamber method (Lapitan et al. 1999).

The amount of N_2O and CH_4 emissions (E_c, $kg\ ha^{-1}$) from paddy fields were computed by multiplying the average daily flux from successive measurements by the number between the sampling dates. The gas samples were analyzed for CO_2, CH_4, and N_2O using gas chromatography (Model 6890N, Agilent Technologies), and the fluxes of GHG gasses were calculated by using Eq. (4) (Zheng et al. 1998).

$$F = \frac{M_w}{M_v} \times \frac{T_{st}}{T_{st} + T} \times \frac{d_c}{d_t} \times h \tag{4}$$

Where, F is the emission fluxes of CH_4, N_2O, or CO_2 ($mg\ m^{-2}\ h^{-1}$); M_w is the molar mass of measured gas ($g\ mol^{-1}$); M_v is the molar volume of measured gas ($L\ mol^{-1}$); T_{st} is the absolute temperature (273.2 K); T is the air temperature at sampling (°C); d_c/d_t is the change in the rate of measured gas concentration ($ppbv\ h^{-1}$); and h is the height of the chamber (m).

2.3.3 Ecosystem service values on C sequestration

The ecosystem service values on C sequestration in this study were calculated by the Sweden C tax (SEPAC State Environmental Protection Administration of China 1997) and afforestation cost in China (Ouyang et al. 1999) by using the following Eq. (5):

$$V_c = \frac{1}{2}\left(C_f + C_t\right) \times E_g \times \frac{12}{44} \tag{5}$$

Where V_c is the ecosystem service values on C sequestration (¥ ha^{-1}); C_f is the mean cost of forestation in China, ¥0.2609 kg^{-1} C (MFPRC Ministry of Forestry of the People's Republic of China 1990); C_t is the Sweden C tax, ¥1.245 kg^{-1} C (SEPAC State Environmental Protection Administration of China 1997); E_g is the cumulative emission or sequestration ($kg\ CO_2\text{-eq}\ ha^{-1}$), including CH_4 and N_2O from paddy fields, net CO_2 emission from plant and soil respiration, and C fixation from photosynthesis. The cumulative emissions of CH_4 and N_2O from paddy fields were converted to CO_2 equivalent using the global warming potential (GWP) factors for a 100-year time horizon, which was converted to CO_2 equivalent by using Eq. (6):

$$E_g = \alpha_{GWP} \times E_c \tag{6}$$

Where E_g is the cumulative GHG emission from paddy fields ($kg\ CO_2\text{-eq}\ ha^{-1}$) including CO_2, CH_4, or N_2O; α_{GWP} indicates the GWP of CO_2, CH_4, and N_2O, which is 1, 25, and 298 for CO_2, CH_4, and N_2O, respectively (Intergovernmental Panel on Climate Change 2013); and E_c is the emission of CO_2, CH_4, or N_2O ($kg\ ha^{-1}$). The CO_2 emissions were the net CO_2 fluxes from plant and soil respiration, because one strain of rice was inclosed within the closed static chamber in the experiment.

2.3.4 Carbon sustainability index

In this study, C sustainability of agricultural system refers to the capacity of reducing GHG emissions and enhancing SOC sequestration to mitigate climate change and sustain agricultural development with low environment costs. Carbon sustainability of a system was assessed by using a holistic approach (Lal 2004a):

$$I_s = \left[\frac{C_O - C_I - C_{OR}}{C_I - C_{IR}}\right]_t \tag{7}$$

Where I_s is the index of carbon sustainability; C_O is the sum of all outputs expressed in CO_2 equivalent including biomass C gain, CH_4, and N_2O emissions from paddy fields; C_I is the sum of all inputs expressed in C equivalent including all agricultural inputs applied; C_{OR} is the output in the reference treatment; C_{IR} is the input in the reference treatment; and t is the time in years. The data of grain yield for the early and late rice for each year were obtained from historical records of the 6-year trial. Biomass production for double rice was computed by using the harvest index (HI) of rice from the published literature (Xie et al. 2011). The C gained in biomass was computed by multiplying the total biomass of rice by its C content (Lin et al. 2006).

3 Results

3.1 GHG emissions in agricultural inputs

Total GHG emissions from agricultural inputs were slightly higher for the late than those for the early rice under different tillage practices, in spite of a higher seeding rate for the early rice but lower fungicides, insecticides, and irrigation (Table 3). With the reduction of tillage frequency and intensity, total emissions from agricultural inputs decreased in the order of CT > RT > NT in both early and late rice (Table 3). Total GHG emissions of agricultural inputs for NT, RT, and CT tillage systems were 1.81, 1.90, and 1.97 Mg CO_2-eq ha^{-1} year^{-1} for the early compared with 1.82, 1.91, and 1.98 Mg CO_2-eq ha^{-1} year^{-1} for the late rice, respectively. With reference to the double paddy cropping systems, total emissions of all inputs for CT were higher than those for RT and NT. With conversion of CT and RT to NT, GHG emissions declined by 0.32 and 0.18 Mg CO_2-eq ha^{-1} year^{-1}, respectively (Table 3).

Overall, the GHG emissions from fertilizers, including urea and compound fertilizer, were the largest contributor to the total emissions of agricultural inputs in double paddy cropping systems (Table 3), accounting for 60.9, 58.0, and 56.0 % under NT, RT, and CT for the early rice and 60.6, 57.7, and 55.7 % for the late rice, respectively. The GHG emission from compound fertilizer, which was ~2.34 times that of urea, was 773.0 kg CO_2-eq ha^{-1} year^{-1} for both the early and late rice in all treatments due to the same amount of compound fertilizer applied in all tillage treatments. Similar to the compound fertilizer, the GHG emissions from pesticides used for each treatment were 38.9 and 57.4 kg CO_2-eq ha^{-1} year^{-1} for the early and late rice, respectively. However, the proportion of pesticides for NT, RT, and CT contributed only 2.1, 2.0, and 2.0 % for the early and 3.2, 3.0, and 2.9 % for the late rice to the total emissions of agricultural inputs, respectively.

There were significant differences in GHG emissions of agricultural inputs among treatments (Table 3). Compared with RT and CT, NT reduced GHG emissions from tillage operation by 89.0 and 159.0 kg CO_2-eq ha^{-1} year^{-1} for the early and late rice, respectively. Regardless of the growing season, the average GHG emission for each treatment was 178.0 kg CO_2-eq ha^{-1} year^{-1} by using the combine harvester. The GHG emission from irrigation were 63.6 and 110.2 kg CO_2-eq ha^{-1} year^{-1} for the early and late rice, respectively. The proportions of GHG emission by machine operation (including tillage, irrigation, and harvesting) were 13.3, 17.4, and 20.3 % for the early rice and 15.8, 19.7, and 22.6 % for the late rice of the total input-based emissions under NT, RT, and CT, respectively.

3.2 Carbon footprint

Regardless of the treatment, the CFs for the early rice were lower than those for the late rice with or without considering the change in SOC stock (Fig. 2). In addition, change in SOC stock had a strong influence on the CF values (Fig. 2). Without considering SOC sequestration, the CF value under NT, RT, and CT were 1.09, 1.36, and 1.26 kg CO_2-eq kg^{-1} year^{-1} for the early rice compared with 1.42, 2.26, and 1.50 kg CO_2-eq kg^{-1} year^{-1} for the late rice, respectively. The CF value for both rice growing seasons were 1.27, 1.85, and 1.40 kg CO_2-eq kg^{-1} year^{-1} for NT, RT, and CT, respectively. Principal contribution to CF was due to CH_4 emission in paddy fields, which was larger for the late than that for the early rice in all tillage treatments. Further, CF values were higher for RT than those for CT and NT in the whole growing season (Fig. 3). But N_2O emissions from paddy fields contributed the least to CF as compared with other emissions for the whole growing season (Fig. 3). In the present study, when SOC changes were accounted for while computing the CF, differences in CF values were observed for each tillage treatment (Fig. 2). The value of CF, including SOC sequestration under NT, RT, and CT, were −0.49, −0.08, and −0.22 kg CO_2-eq kg^{-1} year^{-1} for the early compared with 0.07, 1.07, and 0.27 kg CO_2-eq kg^{-1} year^{-1} for the late rice, respectively. When SOC sequestration was included, the primary contributions to CF were due to CH_4 emission of paddy fields and SOC

Table 3 The GHG emissions of agricultural inputs under different tillage systems in the double rice cropping system (kg CO_2-eq ha^{-1} $year^{-1}$)

Component	Early rice			Late rice			The whole growing season		
	NT[a]	RT	CT	NT	RT	CT	NT	RT	CT
Tillage	0[b]	89.0	159.0	0	89.0	159.0	0	178.0	317.9
Harvest	178.0	178.0	178.0	178.0	178.0	178.0	356.1	356.1	356.1
Irrigation	63.6	63.6	63.6	110.2	110.2	110.2	173.8	173.8	173.8
Fungicide	2.4	2.4	2.4	6.9	6.9	6.9	9.4	9.4	9.4
Herbicide	12.2	12.2	12.2	12.2	12.2	12.2	24.4	24.4	24.4
Insecticide	24.3	24.3	24.3	38.3	38.3	38.3	62.6	62.6	62.6
Urea	329.8	329.8	329.8	329.8	329.8	329.8	659.6	659.6	659.6
Compound fertilizer	773.0	773.0	773.0	773.0	773.04	773.0	1,546.1	1,546.1	1,546.1
Tray	207.4	207.4	207.4	207.4	207.4	207.4	414.7	414.7	414.7
Seed	220.2	220.2	220.2	165.2	165.2	165.2	385.4	385.4	385.4
Total emissions of agricultural inputs (Mg CO_2-eq ha^{-1} $year^{-1}$)	1.81	1.90	1.97	1.82	1.91	1.98	3.63	3.81	3.95

[a] *NT* no-till, *CT* conventional tillage, and *RT* rotary tillage

[b] Emissions from agricultural inputs are the mean value from 2006 to 2011

sequestration for the whole growing season (Fig. 3). The CF from SOC sequestration for the early rice had a greater function than that for the late rice in all tillage treatments, and which for NT was larger than that for CT and RT for both early and late rice seasons (Fig. 3). In contrast to computing CF by excluding SOC, that by including SOC sequestration decreased CF for each tillage treatment for the entire double paddy cropping system.

3.3 Ecosystem services on C sequestration and carbon sustainability

The grain yields under NT, both for early and late rice, were the lowest among tillage treatments from 2006 to 2011 (Fig. 4). In addition, the yield was larger for RT than that for CT for the whole rice season in 2006, and the trends were reversed thereafter (Fig. 4). Considering the function of C sequestration, values of ecosystem services were higher for the early than those for the late rice in all treatments (Table 4). The order of the values of ecosystem services through C sequestration was NT > CT > RT for both early and late rice. The ecosystem service values were ¥2,619, 2,208, and 2,452 ha^{-1} $year^{-1}$ under NT, RT, and CT for the early compared with ¥2,329, 1,145, and 2,110 ha^{-1} $year^{-1}$ for the late rice, respectively. Integrating the values of ecosystem services for the early and late rice, the total value were ¥4,948, 3,353, and 4,562 ha^{-1} $year^{-1}$ for NT, RT, and CT, respectively.

With CT as the baseline, the indices of C sustainability, calculated by Eq. (7), were higher for the late compared with the early rice under NT, but contrary results were obtained for

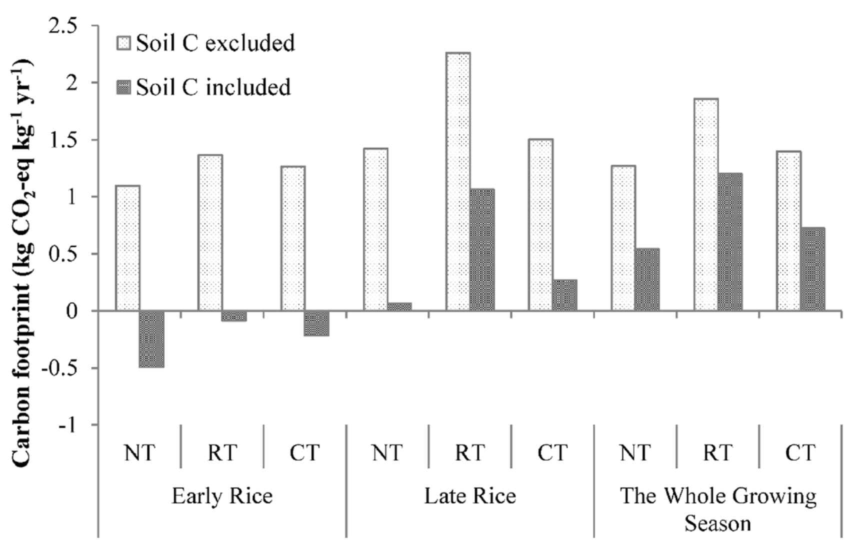

Fig. 2 The carbon footprints (CF) under no-till (NT), conventional tillage (CT), and rotary tillage (RT) in the double rice cropping system

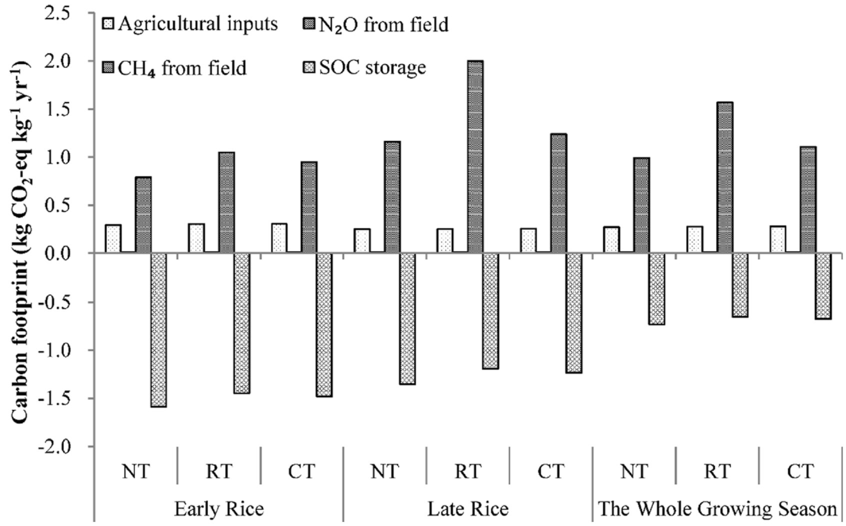

Fig. 3 The component of carbon footprints (CF) sourced from different emissions under no-till (NT), conventional tillage (CT), and rotary tillage (RT) in the double rice cropping system

RT (Table 4). The C sustainability index were 24.42 and 30.00 for the early and 28.07 and −5.84 for the late rice for NT and RT, respectively. Indices of sustainability for the entire growing seasons were 26.25 for NT and −7.91 for RT. Thus, conversion to NT in double paddy fields was more C-friendly than CT. Comparing RT with CT indicated higher C input, lower C output, and lower sustainability index under RT for the entire double paddy cropping system.

4 Discussion

4.1 GHG emissions associated with agricultural inputs

The data presented herein indicate that conversion to NT decreased the total emissions from agricultural inputs by 4.67 and 8.05 % in comparison with RT and CT, respectively. These results are in accord with those of other studies which also reported that conversion of CT to NT reduced GHG emissions from energy and chemicals used in agricultural production (West and Marland 2002; Lal 2004a). In this experiment, the emissions from fuel use for tillage were the main determinants of total GHG emissions among tillage treatments. The differences in fuel consumption among tillage systems were due to differences in tillage frequency, tillage depth, and tractor size. Tillage depth was not uniform among different treatments for both early and late rice seasons. Thus, emission of diesel fuel consumption for CT was higher than that for RT and NT systems. In general, NT can reduce C loss caused by tillage, which can be effective in reducing C emission (Govaerts et al. 2009). Generally, herbicides applied in NT are more than those used in other tillage treatments to control weeds. In order to reduce experimental errors, amounts of agricultural inputs (e.g., fertilizer, seeds, pest, and weed control) were kept the same in all treatments, except

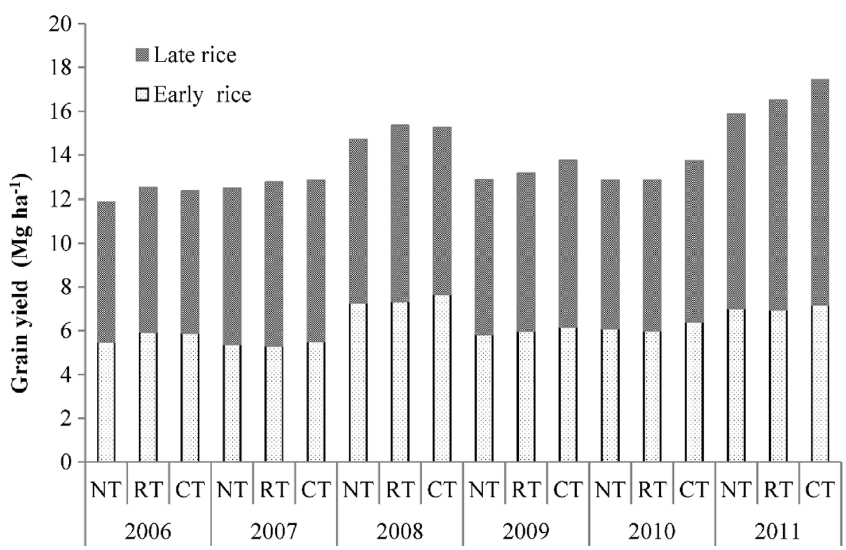

Fig. 4 Grain yields of the double cropping system under no-till (NT), conventional tillage (CT), and rotary tillage (RT)

Table 4 The values of ecosystem service through C sequestration (V_c) and carbon sustainability (I_s) in the double rice cropping system

Source	Early rice			Late rice			The whole growing season		
	NT[a]	RT	CT	NT	RT	CT	NT	RT	CT
Inputs emissions (kg CO_2-eq ha^{-1} $year^{-1}$)	1,811	1,900	1,970	1,821	1,910	1,980	3,632	3,810	3,950
N_2O [b] (kg CO_2-eq ha^{-1} $year^{-1}$)	65	71	79	59	68	60	124	140	139
CH_4 (kg CO_2-eq ha^{-1} $year^{-1}$)	4,894	6,561	6,131	8,425	15,177	9,619	13,318	21,738	15,751
CO_2 (kg CO_2-eq ha^{-1} $year^{-1}$)	−1,662 [c]	−1,224	−1,439	−691	−778	468	−2,353	−2,002	−971
Biomass C gain (kg CO_2-eq ha^{-1} $year^{-1}$)	−17,861	−18,060	−18,681	−20,956	−21,953	−22,402	−38,817	−40,013	−41,083
Vc (¥ ha^{-1} $year^{-1}$)	−2,619[d]	−2,208	−2,452	−2,329	−1,145	−2,110	−4,948	−3,353	−4,562
Is	24.42	30.00	–	28.07	−5.84[e]	–	26.25	−7.91[e]	–

[a] *NT* no-till, *CT* conventional tillage, and *RT* rotary tillage

[b] The N_2O, CH_4, and CO_2 emissions from the field are the mean value from 2006 to 2008. Biomass C gains are the mean value from 2006 to 2011

[c] Negative value means the net fixation of CO_2 from atmosphere to soil or plant

[d] V_c indicates the values of ecosystem services through C sequestration

[e] I_s indicates that the sustainability indexes for RT were better than that for CT due to higher biomass C gain with the lower total GHG emissions

for the tillage practices. Therefore, the actual GHG emission under NT were slightly more than the data presented because of more herbicides used in NT. The quantity of insecticides and fungicides used in this study were computed on the basis of the recommended practices minimizing disease and insect infestation. However, the percentage of GHG emission from herbicides only accounted for a small share, ranging from 0.62 to 0.67 % for the whole growing season (Table 3). Therefore, the trend of GHG emission in NT of this study would not change because of the small percentage of total emissions in comparison with that from the local farming practices for NT.

The data presented herein showed that total GHG emissions of agricultural inputs in the early rice were slightly lower than those from the late rice. These differences can be attributed to different levels of agricultural inputs in two seasons. In general, higher seeding rates were used to ensure a good seedling emergence due to lower temperature in the spring for the early rice, which led to more GHG emissions by seed input than that in the late rice. However, higher temperature and risks of drought were observed during the late than for the early rice, thus leading to more serious disease and insect infestation. Therefore, more pesticides were applied for the late than for the early rice, resulting in more GHG emission in the late than in the early rice (Table 3). Meanwhile, the GHG emission from irrigation was larger for the late rice than that for the early rice in the study. The larger irrigation amount for the late rice was due to the less precipitation during the late rice season in comparison with the early rice season. The irrigation water was pumped by a diesel-powered pump in the study. However, using the artesian water through canals and ditches from a nearby artificial reservoir for irrigation is more common in the region, which could save the diesel consumption compared to our study. Thus, the GHG emission from irrigation in this study could be higher than the actual level in the experimental region. Estimates of the GHG emission presented herein are lower than the actual emissions because of the lack of information about the emission in CLCD dataset for the production process involving the use of tray for growing seedlings. Above analyses on inventory data will influence the accuracy of results to some extent for GHG emissions of agricultural inputs, but the trend remains unchanged among different tillage systems.

4.2 Soil organic carbon sequestration and carbon footprint

The SOC sequestration, due to changes in land use and management (e.g., soil tillage and residue retention), is one of several options of reducing net anthropogenic emission of CO_2 and mitigating global warming. Long-term use of NT and the retention of crop residues on cropland can strongly influence soil properties and environment quality while maintaining and enhancing SOC concentration (Kahlon et al. 2013). Changes in SOC stock can strongly influence CF. The data presented herein show that the CF value decreased significantly when change in SOC storage was taken into account, as was also reported in other studies (Gan et al. 2012a). Consideration of SOC stock in the computation of CF can strongly alter the values, sometimes from positive to negative (Gan et al. 2012a). The negative value of CF indicates a net C sink capacity of a cropping system. In the present study, the CF including SOC sequestration was converted to a negative value in the early rice under all treatments. Yet, the CF value remained positive for all tillage treatments in the late rice. This trend may be due to larger CH_4 emissions from paddy fields in the late rice (Fig. 3). The below-ground biomass C is a principal source of SOC. Further, soil depth for computing the SOC stock was insufficient for the

assessment of C sequestration because crop roots generally extend to deeper than 30 cm (Baker et al. 2007). However, lower soil temperature and higher penetration resistance in NT system may affect root growth to deeper layers and their distribution in the entire soil profile (Baker et al. 2007). This trend would lead to no or less SOC sequestration in subsoil under NT than that in the CT system. By considering SOC sequestration in deeper layers, the CF value could increase under NT but decrease under CT. Most studies thus far have selected 100 years as the time scale for assessing the GWP (Xiao et al. 2005; Gan et al. 2012a, b, c). Therefore, the CO_2 equivalent of the amount of non-CO_2 GHG emissions (CH_4 and N_2O) were calculated as the mass of the specific GHG multiplied by its GWP based on 100-year time. Thus, management practices would be valid for this time after the conversion of CT or RT to NT system. However, management practices in rice production system could change during that time, which would alter the CF value. Regardless of SOC, the CF values were in the order of RT > CT > NT in the double rice cropping system. The lowest emissions from agricultural inputs in paddy fields and the largest change in SOC storage under NT are the possible reasons of differences in CF values among tillage systems. Yet, additional research is needed to ascertain the merits of including management-induced changes in SOC stock in the computation of CF for grain crop production systems (Gan et al. 2012a). The protocol used for CF computation must be standardized.

4.3 Ecosystem services on C sequestration

A multitude of goods and services are derived from soil as the natural capital and ecosystem service provisioned by it (Costanza et al. 1997; Lal 2013). Agro-ecosystems can create numerous important ecosystem services, such as food, feed, fiber, and climate regulation (Kroeger and Casey 2007). Therefore, ecosystem services can be an integrated indicator for the assessment of sustainability. It is also important to realize that C sequestration plays an important role in ecosystem services by reducing GHG emissions. Therefore, enhancing ecosystem services, particularly through C sequestration, is important to mitigating climate change. The ecosystem service value of C sequestration was only assessed as a specific ecosystem service to interpret C sustainability from an economic viewpoint without consideration of other service functions provisioned by paddy fields in this study. The data presented herein show that the values of ecosystem services on C sequestration for the early rice were higher than those for the late rice, because of lower CH_4 emission and higher CO_2 uptake in the early rice. On the whole, conversion to NT can improve the value of ecosystem services by reducing the GHG emission and enhancing SOC sequestration for the entire seasons. Indeed, ecosystem services are very complex and link economic, ecologic, and the human dimensions which make it difficult to assess the value of total ecosystem services. In addition, this study mainly focused on the functions of SOC sequestration by the conversion of conventional farming practices to CA. Therefore, we only assessed the ecosystem services based on C sequestration, without considering the total services of an ecosystem. Yet, strong links exist among diverse ecosystem services. Therefore, relationships between C sequestration and other ecosystem services would be assessed in details in future studies (Lal 2013).

Among diverse cropping systems practiced in the region (e.g., wheat-rice system and the double paddy system), relatively large accumulation of biomass C has been reported in double rice cropping than that in the single season rice (Xiao et al. 2005). In addition, the emission of GHG from paddy fields and those from agricultural inputs are also computed in the present study. Here, although the site experiment was located in the typical double rice cropping area, due to soil and spatial heterogeneity, there are still some uncertainties in the assessment of GHG emission from the field for a regional scale of double rice cropping system in China. While the datasets for agricultural inputs were collected from the site experiment, there may be an inconsistency among different regions due to diversity in farm management practices. Thus, more studies and investigations need be done to gain scientific information on GHG emissions in those areas with the double rice cropping system. The Swedish C tax was applied for assessing ecosystem services on C sequestration due to scarcity of the Chinese C tax, which will influence the accuracy of results due to the differences of agriculture technology and inputs. In spite of this limitation, the general trends of ecosystem services on C sequestration did not vary among treatments. In another study on assessment of ecosystem services in China, Ouyang et al. (1999) also adopted the parametric approach used in the Swedish C tax assessment. However, there exists a strong need to develop and improve the database of C tax and LCA for China, which has important polity implication. The accuracy of the eco-service value can be improved by the assessment of the value of the C tax in future studies.

5 Conclusions

The data presented herein showed that total GHG emissions from agricultural inputs ranged from 3.63 to 3.95 Mg CO_2-eq ha^{-1} $year^{-1}$ for the double cropping rice system, and most of these emissions were attributed to fertilizer input. The CF of double paddy was reduced sharply by including changes in SOC stock. Upon conversion of RT and CT to NT, the value of ecosystem services on SOC sequestration increased by ¥1,595 and 386 ha^{-1} $year^{-1}$, respectively. Lower CF was observed for the early compared with the late rice for all tillage treatments.

The relative order for both the early and late rice was NT < RT < CT for GHG emission, NT < CT < RT for CF, and NT > CT > RT for ecosystem services on C sequestration. With CT as the baseline, the C sustainability index was better for NT than for RT for the entire double paddy cropping system. Thus, even the short-term NT is a more C-friendly and sustainable technique for mitigating GHG emissions, improving ecosystem services and SOC sequestration in the double rice production system. However, additional research is needed to improve the NT system for enhancing grain yield and agronomic productivity.

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